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Journal of Applied Ecology

Journal of Applied Ecology 2016 doi: 10.1111/1365-2664.12784

Developing a biodiversity-based indicator for large-scale environmental assessment: a case study of proposed shale gas extraction sites in Britain

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Summary

1. Environmental impact assessments are important tools for predicting the consequences of development and changes in land use. These assessments generally use a small subset of total biodiversity – typically rare and threatened species and habitats – as indicators of ecological status. However, these indicators do not necessarily reflect changes in the many more widespread (but increasingly threatened) species, which are important for ecosystem functions. In addition, assessment of threatened species through field surveys is time-consuming and expensive and, therefore, only possible at small spatial scales. In contrast, planning changes in land use over large spatial scales (e.g. national infrastructure projects) require assessment and prioritization of biodiversity over large spatial extents.

2. Here, we provide a method for the assessment of biodiversity, which takes account of species diversity across larger spatial scales, based on occurrence records from 5553 species across 11 taxonomic groups. We compare the efficacy of the biodiversity-based indicator we developed against one based on threatened species only and then use it to consider spatial and temporal patterns in ecological status across Great Britain. Finally, we develop a case study to investigate biodiversity status in regions proposed for shale gas extraction in Great Britain.

3. Our results show a strong relationship between the ecological status of areas defined by all biodiversity versus only threatened species, although they also demonstrate that significant exceptions do exist where threatened species do not always accurately indicate the ecological status of wider biodiversity.

4. Spatial and temporal analyses show large variation in ecological status across Great Britain both within the area made available for shale gas licensing and within individual environmental zones. In total, however, 63% of hectads across Britain have suffered a net reduction in our biodiversity-based indicator since 1970.

5. *Synthesis and applications*. We provide a method and develop a biodiversity-based indicator for the assessment and prioritization of biodiversity at large spatial scales. We highlight the potential applications of this approach for the prioritization of areas that would benefit from conservation and restoration. We also emphasize the danger of insufficient consideration of more widespread species and not just rare and threatened species and habitats as indicators of ecological status when prioritizing large-scale national infrastructure projects. Our method should be a useful tool to complement existing environmental impact assessment methods.

Key-words: biodiversity indicators, biodiversity prioritization, ecological status, ecosystem function, environmental impact assessment, fracking, priority species, shale gas, shale gas extraction, strategic environmental assessment

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Introduction

The quantification and prioritization of biodiversity is a major challenge for conservation biologists and policy-makers (Balmford *et al.* 2005). Increasing pressures on land use including food and energy security and housing development are resulting in a growing need for tools to identify and prioritize areas of 'high' ecological value (Mace 2005; Scholes & Biggs 2005; Yoshioka, Akasaka & Kadoya 2014). There are various tools based on the extent of land cover which might be used as proxy for biodiversity, but these make a number of assumptions and either show limited fit to empirical data, or have not yet been appropriately validated (Willis *et al.* 2012; Terrado *et al.* 2016).

In Great Britain (GB), environmental impact assessments (EIAs) and strategic environmental assessments (SEAs) are currently used to predict the environmental consequences of changes in land use on potential development sites (Slootweg & Kolhoff 2003; Kolhoff *et al.* 2009). Major shortfalls of these assessments include the low priority given to biodiversity generally, and the focus on a small subset of priority species and habitats (Treweek 2001; Rajvanshi, Mathur & Slootweg 2009). Biodiversity is typically assessed using threatened species, threatened habitats and Sites of Special Scientific Interest (SSSI) as indicators of ecological status, for example using the Biodiversity Action Plan (BAP; Joint Nature Conservation Committee 2012) which defined priority species and priority habitats in GB up to 2010, before these lists became devolved to separate countries (i.e. Scotland, England and Wales). Whilst the use of these indicators can prove to be effective for prioritizing and conservation at a local level (e.g. Treweek 2004), they only represent a small proportion of overall biodiversity and may not necessarily reflect spatial patterns and temporal trends in this 'wider' biodiversity (Franco *et al.* 2009). For example, requirements of species such as the Great Crested Newt *Triturus cristatus,* which is a European Protected Species listed under the Habitats Directive, do not necessarily match those of other species which might benefit from certain interventions such as development of green infrastructure (Van Teeffelen *et al.* 2015). Beyond the small subset of legally protected species, many previously widespread species are in decline; therefore, reporting on their status is important (Burns *et al.* 2013), especially as these common species may underpin crucial ecosystem functions (Winfree *et al.* 2015).

The restricted capacity of threatened species as indicators of biodiversity is particularly relevant for large-scale projects or policies (e.g. national energy infrastructure), which have the potential to impact biodiversity across large areas comprising multiple ecosystem types. Essentially, the current implementation of EIAs and SEAs does not consider impact at the ecosystem level, which is critical for conservation and planning/policy decisions at both regional and national scales (Gontier, Balfors & Mörtberg 2006; Lawton *et al.* 2010; HM Government 2011). For example, the UK government is facilitating the exploration of shale gas deposits and has opened large extents (44·8%) of land area for exploration licences, with plans to open up more land in the near future (UK Department of Energy and Climate Change). Whilst environmental assessment of this land identifies priority species/habitats and SSSIs that may be affected (Moore, Beresford & Gove 2014), ecological impacts on wider biodiversity, which may be significant at a national level, are not considered. Other national-scale projects and policies (e.g. rail linkages such as the planned HS2 high speed line in GB, and also wind farm allocation, Bakken *et al.* 2014), follow the same pattern, highlighting the crucial need for improved indicators for biodiversity assessment at larger scales. Scientifically robust, consistent and readily accessible indicators are especially important, as they should provide the preliminary step in the prioritization of land at a large spatial scale before more detailed and costly local assessments. Preventative action at this wider scale may limit reductions in ecosystem services caused by losses of local biodiversity (Balvanera *et al.* 2006; Cardinale *et al.* 2012). In addition to identifying the potential impact of development, governments may wish to identify areas for restoration in order to improve the status of biodiversity (e.g. in accordance with the 2020 Aichi targets set by the Convention for Biodiversity; http://www.cbd.int/sp/targets/). Also potential biodiversity offsetting initiatives required information on spatial patterns in biodiversity in order to design appropriate compensation measures.

Here, we use a state-of-the-art method to analyse species occurrence data, of the kind often collected by volunteer recorders (citizen scientists) on behalf of national species recording schemes, for the quantification and prioritization of biodiversity across Great Britain (Pocock *et al.* 2015). These data represent a valuable resource for land prioritization based on ecological value (c. 111 million records in the UK, see NBN Gateway; https://data.nbn.org.uk/, accessed July 2015). However, uneven sampling at temporal and spatial scales can cause uncertainty in the statistical analysis of occurrence data (Boakes *et al.* 2010; Isaac & Pocock 2015). In order to account for any spatial variation in recorder effort, we use 'Frescalo' (Hill 2012), a recently developed method that standardizes for recorder effort and allows the analysis of relatively unstructured occurrence data. Previous uses of Frescalo have been restricted to individual taxonomic groups (e.g. bryophytes, Hill 2012; and moths, Fox *et al.* 2014). In order to create a biodiversity-based indicator of ecological status, we use data from 11 well recorded taxonomic groups (representing 5553 species in total) recorded between 1970 and 2013 and stratify our analysis by abiotic variables that are likely to be associated with biodiversity. We investigate spatial and temporal patterns of this indicator and apply the method to consider the ecological status of areas opened for shale gas exploration licences in Great Britain.

Our work has three aims: first, to develop a scientifically robust and consistent indicator of ecological status (species richness of a location relative to the potential maximum for a given abiotic context), which takes account of wider biodiversity beyond threatened conservation 'priority' species; secondly, to compare spatial and temporal trends in ecological status based on priority species and on overall species diversity (i.e. including non-priority species); and thirdly, to demonstrate the utility of this novel tool by applying it to quantify biodiversity patterns in the area proposed for shale gas licensing.

Materials and methods

DATA COLLATION

Species occurrence data were collated from the Biological Records Centre (on behalf of several national recording schemes), the British Trust for Ornithology and Butterfly Conservation. These biological records are checked for quality by expert local co-ordinators before submission to national data bases. Species concepts are those currently recognized by scheme co-ordinators, but in some cases, such as vascular plants, future taxonomic revisions involving aggregation of subspecies or disaggregation of species could be possible. Data were gathered for 5553 species across Great Britain (GB) for 11 taxonomic groups at the 10 km x 10 km square scale (hectad) (Table 1). These comprise primarily terrestrial species, and, although they are likely to cover a broad range of habitats and functional roles, we do not presume they are fully representative of all GB ecosystems. However, they do represent the most comprehensive analysis of spatial biodiversity patterns across an entire country to date, and in taxonomic breadth, they are a large improvement over considering only single well-studied groups (e.g. birds and butterflies). Taxonomic groups were selected where their data covered more than 50% of the total hectads sampled across Great Britain for the two time periods analysed (discussed subsequently). The threshold was applied to ensure sufficient sampling coverage in order to maximize the accuracy of species richness estimation using the Frescalo method. In order to investigate trends over time, species richness in each taxonomic group was assessed in two time intervals separated by 10 years: 1970–1990 and 2000–2013. An earlier baseline was not chosen as the quality and quantity of available biological record data before 1970 were insufficient for our analysis. Given this earliest possible baseline, we then selected the two time periods in order to balance the need to have sufficient data within each period in order to maximize statistical power of the Frescalo method, versus maintaining as large a time gap between them as possible in order to detect a signal of species richness change. For birds, we used data corresponding to the two time periods defined above, taken from the atlases of 1968–1972 (Sharrock 1976) and 2007–2011, respectively (Balmer *et al.* 2012). For vascular plants, non-native species were excluded from our analysis, due to the large percentage of non-natives in this taxonomic group in GB (Roy *et al.* 2014). Many of these represent escapes from gardens, and from a biodiversity conservation perspective, it is appropriate to exclude them from the indicator. However, additional analysis (not shown) found that the overall results for aggregate ecological status across all groups were qualitatively similar with non-native plants included.

ESTIMATING SPECIES RICHNESS

Analyses were undertaken separately for both time periods for each taxonomic group, on the basis that the biological recorders tend to focus on a specific taxon (e.g. see Acknowledgements for the list of recording schemes and societies). In each analysis, we compiled a species list and calculated the observed 'raw' species richness for each hectad, and then, we applied Frescalo (Hill 2012) to account for the variation in recorder effort within different hectads. The Frescalo program estimates species richness at a given location according to the set of species occurring in a neighbourhood of the 100 most similar hectads from the 200 nearest hectads. The proportion of a suite of common benchmark species from this neighbourhood list that have been recorded in the focal

Table 1. Summary of data used to produce indicators of ecological status

Data set	Hectad occurrence (%; 1970–1990)	Number of spp. (1970–1990)	Priority spp. (1970–1990)	Hectad occurrence (%; 2000–2013)	Number of spp. (2000–2013)	Priority spp. (2000–2013)
Bees	52	227	60	70	231	59
Birds*	100	233	106	100	287	112
Bryophytes	90	1087	275	91	1161	277
Butterflies	93	59	24	98	59	23
Carabids	82	351	30	50	355	34
Hoverflies	82	270	18	81	266	19
Isopods	86	47	0	55	50	0
Ladybirds	51	49	0	59	52	0
Macromoths	85	980	134	90	977	135
Grasshoppers and crickets	70	55	7	52	74	7
Vascular plants (native to GB)	99	1886	295	84	1860	288

Data include species occurrence records from 11 taxonomic groups, collected over two distinct time periods: 1970–1990 and 2000–2013. The total number of species and the number of priority species only are summarized for each group in both time periods. Only taxonomic groups with over 49·6% GB coverage were included.
*Data taken from Bird atlases of 1968–1972 and 2007–2011, respectively.

hectad is used to assess the recording intensity of this focal hectad, and this is used to scale the 'raw' observed species richness towards the neighbourhood maximum accordingly. We ran Frescalo using the *Sparta* package (August, Harrower & Isaac 2013) in the program R (R Core Team 2013). Neighbourhoods were defined according to biological similarity using either vascular plant data or land cover type: all data sets were analysed using the vascular plant weights files embedded within the Frescalo program, with the exception of the vascular plants data set for which biological similarity between hectads was defined using land cover type in order to avoid circularity. For this, we used the 2007 ITE Land cover Map (Morton *et al.* 2011).

The Frescalo method has been tested and validated and shown to be a robust method for estimating species occurrences by accounting for spatiotemporal variation in recorder effort. This validation has comprised a) repeating the method with subsets of a well-sampled data set, during development of the method for bryophytes (see Supporting Information of Hill 2012, figs S1–S6, tables S1–S4); b) comparison to 'raw' unstructured occurrence data during an analysis of moths (Fox *et al.* 2014); and c) comparison to simulated occurrence data where recorder effort and the location of true 'absences' is known; the method performed very well compared to other methods to account for recorder effort (Isaac *et al.* 2014). In addition, in the current study, we carried out further validation for one of the taxonomic groups, butterflies, for which independent abundance monitoring data were available. We compared Frescalo-estimated species richness from the butterfly occurrence data set with species richness from the UK Butterfly Monitoring Scheme (UKBMS), which uses a standardized transect methodology with up to 26 repeat visits per year. The set of raw species occurrence records for butterflies for 2000–2013 were 'degraded' by randomly removing 20, 50 or 80% of records in order to assess the effects of recording intensity on the accuracy of species richness estimates. Species richness within each degraded data set was estimated either using the Frescalo method or with no control for recorder effort and then plotted against observed species richness from the UKBMS data for matching hectads. As the UKBMS transects only sample a fraction of a given hectad (transects tend to be linear or circular routes of around 1–3 km in length and 5 m wide), we would always expect the species richness recorded in the UKBMS to be lower than the 'true' species richness of the hectad. However, comparing controlled (Frescalo) and uncontrolled (raw data) species richness against UKBMS species richness does allow assessment of the relative degree of under-recording in each.

MEASURING ECOLOGICAL STATUS

Our aim was to collate information across many species groups to produce an indicator which reflects the ecological 'quality' of a hectad (i.e. in which the detrimental effects of land use which are harmful to biodiversity are absent). A comparison of raw species richness over large spatial scales (e.g. different regions in a country) is, therefore, not necessarily a useful measure of this ecological 'quality' (hereon referred to as ecological status) due to the different species assemblages associated with differing abiotic conditions that cannot be manipulated by ecosystem managers, for example the strong latitudinal gradient in climate in Britain. We therefore used a relative measure of estimated species richness to calculate the ecological status of each hectad. Abiotic conditions were taken into account by assigning each hectad to an environmental zone. For this, we used the ITE land classification (Bunce *et al.* 1996, see Table S1 (Supporting Information) which classifies areas using a combination of land cover type, climate, geology and topography. We assigned zones according to the dominant ITE land class (45 classes in total) present in individual hectads (see Fig. S1). The estimated species richness for any given hectad was then compared as a proportion of the total species richness in the most species-rich hectad of the relevant environmental zone, in order to give an ecological status 'score' for that taxonomic group. This effectively quantifies how species rich a focal hectad is relative to how species rich it *could* be given the abiotic conditions in that environmental zone. We recognize here that, potentially, even the most species-rich reference hectad could already be degraded. However, given that this is the first time that multitaxa species richness has been quantified in GB, we have no choice but to pragmatically accept this as our starting baseline from which to measure further change, with the caveat that it may be an underestimate of the true potential species richness of an environmental zone. Therefore, we calculated ecological status from the latter time period (2000–2013) relative to the species richness maximums from the earlier time period (1970–1990), chosen as our historical baseline. Overall spatial and temporal trends in ecological status were calculated through comparison of the mean ecological status, taken across all taxonomic groups, in each of the defined time periods. Hence, in our analysis, each taxonomic group is given equal weight in contributing to the indicator although future work could alter these weightings if certain groups were regarded as more important, for example for particular ecosystem services.

COMPARISON BETWEEN THE ECOLOGICAL STATUS OF WIDER BIODIVERSITY AND GB 'PRIORITY' SPECIES

We compared the ecological status derived from the wider GB biodiversity (mean ecological status across all taxonomic groups; 5553 species; Table 1) with the ecological status derived from GB priority species (955 species). We used an updated list of GB priority species adapted by the Joint Nature Conservation Committee from the BAP 2007 species list and reflecting all species on the new devolved priority species lists for England, Scotland and Wales (Joint Nature Conservation Committee 2012). For this subset of all GB species, we derived ecological status using the same method described previously. We used a linear regression analysis to test for the correlation between the ecological status of wider biodiversity and priority species for all hectads within GB.

APPLICATION OF METHODS AS A PRIORITIZATION TOOL

Spatial and temporal patterns in the ecological status of both wider biodiversity and of priority species were compared at large (across environmental zones) and small (within environmental zones) spatial scales. The ecological status of the most recent time period was used to assess the area recently offered for shale gas licensing (data obtained from the UK Department of Energy and Climate Change). We explored spatial trends and the status of both the wider biodiversity and of priority species for this area.

Results

FRESCALO ANALYSIS AND ADDITIONAL VALIDATION USING UK BUTTERFLY MONITORING SCHEME DATA

The number of species recorded for each taxonomic group varied from 47 species for the Isopoda to 1886 species for the vascular plants. The minimum proportion of hectads sampled across GB across all 11 taxonomic data sets was 50% (carabid beetles) and over 80% in six of the data sets analysed (Table 1). The analysis of these species occurrence data in Frescalo provided estimates of species richness for each hectad within each taxonomic group (e.g. Fig. 1; see Dyer & Oliver 2016 for data). Comparisons of Frescalo output for butterflies with the UKBMS transect data showed the Frescalo method to reduce underestimates of species richness effectively, even when the observed data were degraded by removing up to 80% of original species records (Fig. S2). This empirical validation was unable to detect potential overestimation of species richness, but recent simulation analyses have also shown Frescalo to be a robust method for estimating species occurrence and trends over time (Isaac *et al*. 2014). Despite the extensive validation of Frescalo (Hill 2012; Fox *et al*. 2014; Isaac *et al*. 2014), it should be remembered that it is a probabilistic estimate of species richness based on surrounding similar hectads rather than an absolute measure. Perfect species richness estimates are unlikely because focal hectads could still differ slightly in their quality for biodiversity despite being close-by and having similar botanical or land cover composition. Nevertheless, validation shows that Frescalo method is a vast improvement on using 'raw' unstandardized species richness estimates, and we believe it is a valid approach as a preliminary screening tool for environmental assessment.

SPATIAL AND TEMPORAL TRENDS IN ECOLOGICAL STATUS OF THE WIDER GB BIODIVERSITY

Ecological status was calculated as the proportion of total species richness in a given hectad relative to the most species-rich hectad in the associated environmental zone in 1970–1990. Spatial and temporal changes in the ecological status of the wider GB biodiversity (taken as the mean from across the 11 taxonomic groups) between the two time periods are shown in Fig. 2. The results are also shown on an interactive online application (https://eip.ceh.ac.uk/apps/ecostatus/).

Between 1970 and 1990, the mean ecological status across all hectads and environmental zones was $0{\cdot}71 \pm 0{\cdot}002$ (i.e. the average hectad contained 71% of the total number of species observed in the most species-rich hectad, given abiotic conditions; Fig. 2a), and between 2000 and 2013, the mean ecological status was $0{\cdot}70 \pm 0{\cdot}002$ (Fig. 2b). The proportion of hectads with 'high' ecological status (>0·8; arbitrarily decided according to the observed mean ecological status) was 21·3% and 19·3%, respectively, in the two time periods and the proportion of hectads with 'low' ecological status (<0·6; arbitrarily decided according to the observed mean ecological status) was 15·3% and 18·3%, respectively.

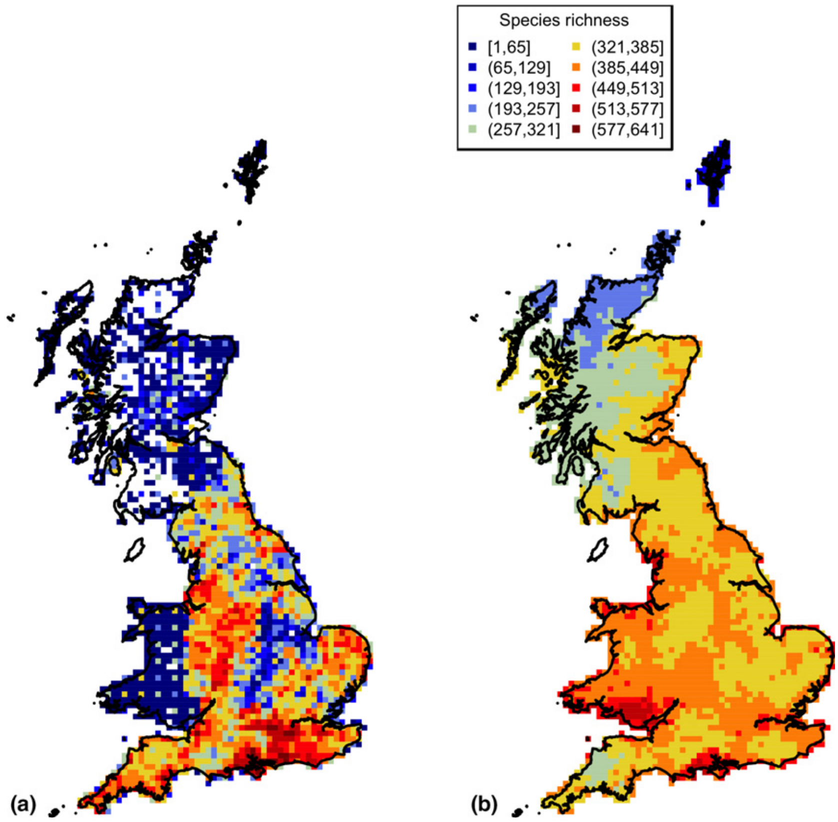

Fig. 1. Species richness and estimated species richness per hectad for vascular plants in GB. Panel (a) shows raw recorded species richness. National boundaries and a standardized national plant survey (gridded pattern) can be identified – both patterns that are artefacts of spatial variation in recorder effort. Panel (b) shows estimated species richness obtained using the Frescalo program (Hill 2012) to standardize for recorder effort.

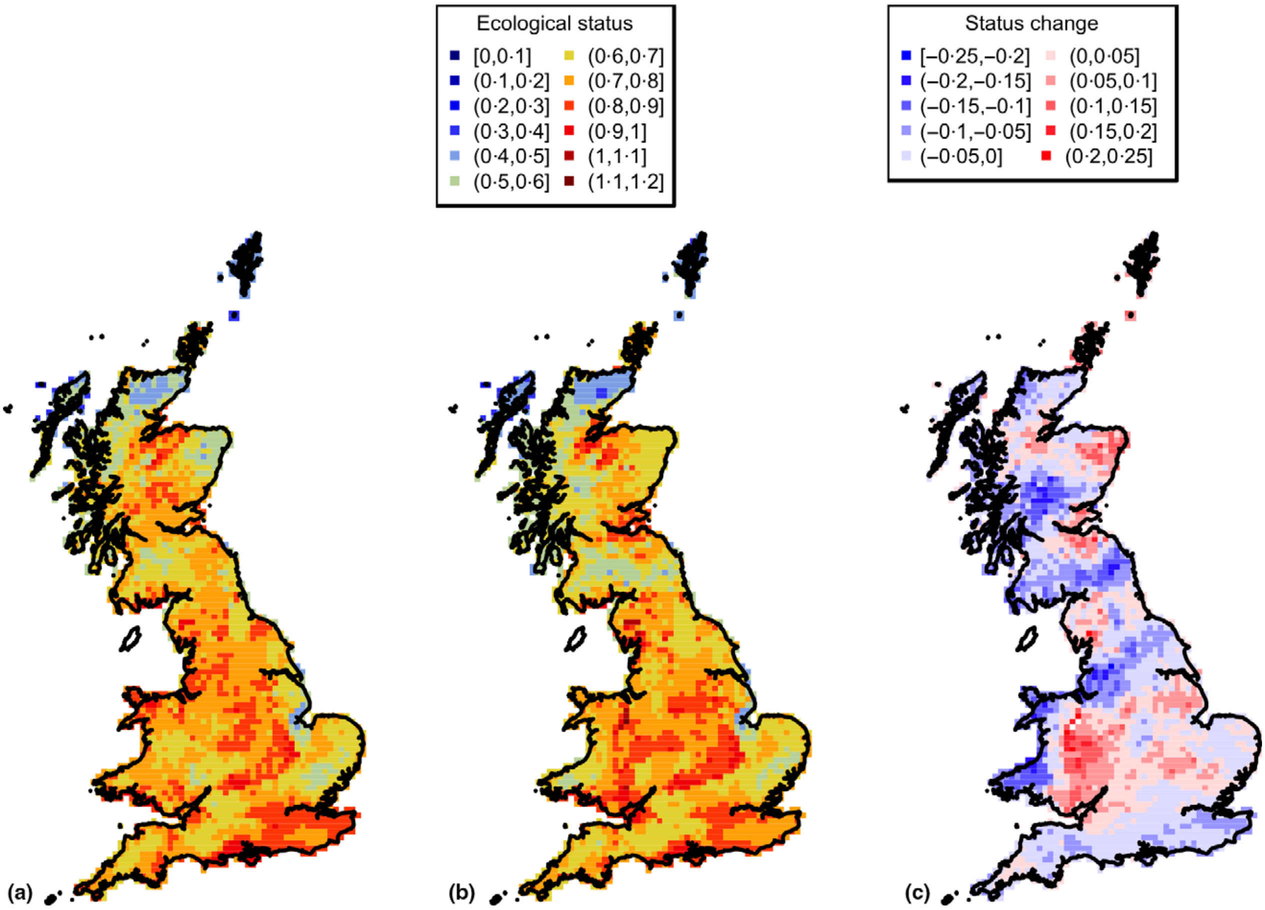

Fig. 2. Spatial and temporal patterns of biodiversity in GB. (a) Map showing spatial patterns of mean ecological status, calculated from relative species richness estimates (according to maximum species richness observed under similar abiotic conditions) for all taxonomic groups between 1970 and 1990; (b) map showing spatial patterns in mean ecological status calculated from relative species richness estimates (relative to maximum species richness observed in the 1970–1990 time period) for all taxonomic groups between 2000 and 2013; (c) map showing temporal change in ecological status between the two time periods (1970–1990 and 2000–2013).

Smaller scale analysis of spatial patterns within individual ecological zones showed considerable variation. In the most recent time period, more than half of the hectads in five environmental zones had ‘high’ ecological status, and more than half of the hectads in three zones had ‘low’ ecological status. Environmental zones 11e (flat plains/small river floodplains, E Midlands) and zone 3e (flat/gently undulating plains, E Anglia/S England) are used to illustrate this variation (Fig. 3). In zones of low ecological status (e.g. zone 3e; Fig. 3b), spatial patterns can be used to identify areas which might be prioritized for restoration to improve connectivity.

Analysis of the temporal change in ecological status over the two time periods identified an overall decrease in ecological status (Fig. 2c). Ecological status decreased in 63% of hectads (1738 out of 2799; proportion test: $\chi^2 = 163{\cdot}3$, $P < 0{\cdot}001$). Furthermore, 192 (7%) of the hectads showed a ‘large’ decrease in ecological status (defined as a change >0·1, i.e. >10% of the total number of species observed in the most species-rich hectad in the environmental zone were lost over time), whilst 99 (3·5%) of the hectads showed a large increase in ecological status (comparison of the two proportions: $\chi^2 = 30{\cdot}7$, $P < 0{\cdot}001$). A decrease in ecological status was observed in 33 environmental zones, and an increase in ecological status was observed in 12 environmental zones (Fig. 4a; $\chi^2 = 17{\cdot}8$, $P = <0{\cdot}001$).

Large increases or decreases in ecological status over time for individual hectads can be investigated by observing the underlying ecological status data for the comprising taxonomic groups. For example, a large increase in ecological status (0·25) was observed in hectad SJ12 in environmental zone 17W1. Disaggregation of the mean ecological status showed that seven groups increased and four groups decreased in ecological status over time, but there was a very large increase in the ecological status of bees (see Table S2). Hectad SD83, in environmental zone 18e, showed a large decrease in ecological status over time (−0·20). Here, the ecological status of the individual taxonomic groups increased in three groups and decreased in eight groups, with the largest declines observed in grasshoppers and crickets, bees and hoverflies (see Table S3).

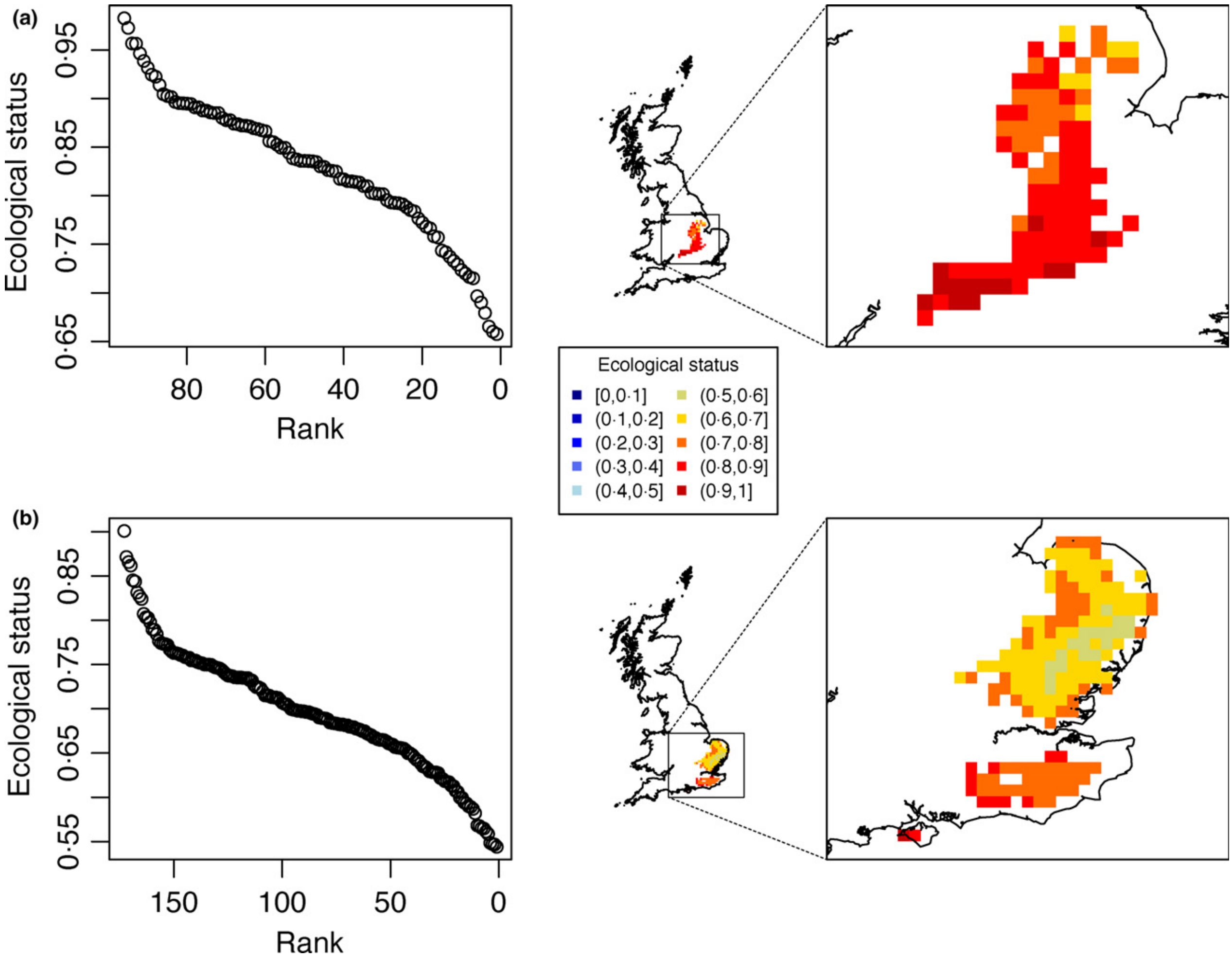

Fig. 3. Assessment of GB biodiversity on a smaller scale: patterns of ecological status (between 2000 and 2013) within two individual environmental zones. (a) An example of an environmental zone (11e; flat plains/small river floodplains) with many hectads of 'high' ecological status (mean = 0·83), including a scatter plot of ranked ecological status across all hectads in the zone and a map illustrating the spatial patterns of ecological status. (b) An example of an environmental zone (3e; flat/gently undulating plains, E Anglia/S England) with many hectads of 'low' ecological status (mean = 0·70), including a scatter plot of ranked ecological status across all hectads in the zone and a map illustrating the spatial patterns of ecological status. This map could be used to identify areas where connectivity may be improved to create larger areas of high ecological status.

THE RELATIONSHIP BETWEEN THE ECOLOGICAL STATUS OF PRIORITY SPECIES AND WIDER BIODIVERSITY

A strong positive correlation was observed between the ecological status score based on all species (mean ecological status of 11 taxonomic groups) and the ecological status of priority species (Fig. S3; $P < 0{\cdot}001$, $R^2 = 0{\cdot}50$). Although this showed that ecological status of priority species does generally reflect that of wider biodiversity, we do note some exceptions (Fig. S3). These show that the species richness of priority species as an indicator can sometimes overestimate or underestimate ecological status estimated for wider biodiversity. Caution should always be taken when comparing estimates for individual hectads, given uncertainty as in any statistical methodology. Importantly, however, some whole environmental zones showed opposing temporal trends in ecological status estimated from wider biodiversity versus priority species, suggesting that differences may well be genuine even given the uncertainty in individual hectad estimates (Fig. 4).

The mean ecological status of hectads derived from priority species was $0{\cdot}79 \pm 0{\cdot}002$ between 1970 and 1990, and $0{\cdot}74 \pm 0{\cdot}002$ between 2000 and 2013. The proportion of hectads with 'high' ecological status was 51·5% and 30·9% in the two periods, respectively ($\chi^2 = 245{\cdot}3$, $P < 0{\cdot}001$), and the proportion of hectads with 'low' ecological status was 5·2% and 8·9%, respectively ($\chi^2 = 29{\cdot}6$, $P < 0{\cdot}001$). Ecological status of priority species decreased in 76·6% of hectads ($\chi^2 = 789{\cdot}8$, $P < 0{\cdot}001$), and showed large decreases (>0·1) in 24·7% of hectads ($\chi^2 = 715{\cdot}1$, $P < 0{\cdot}001$).

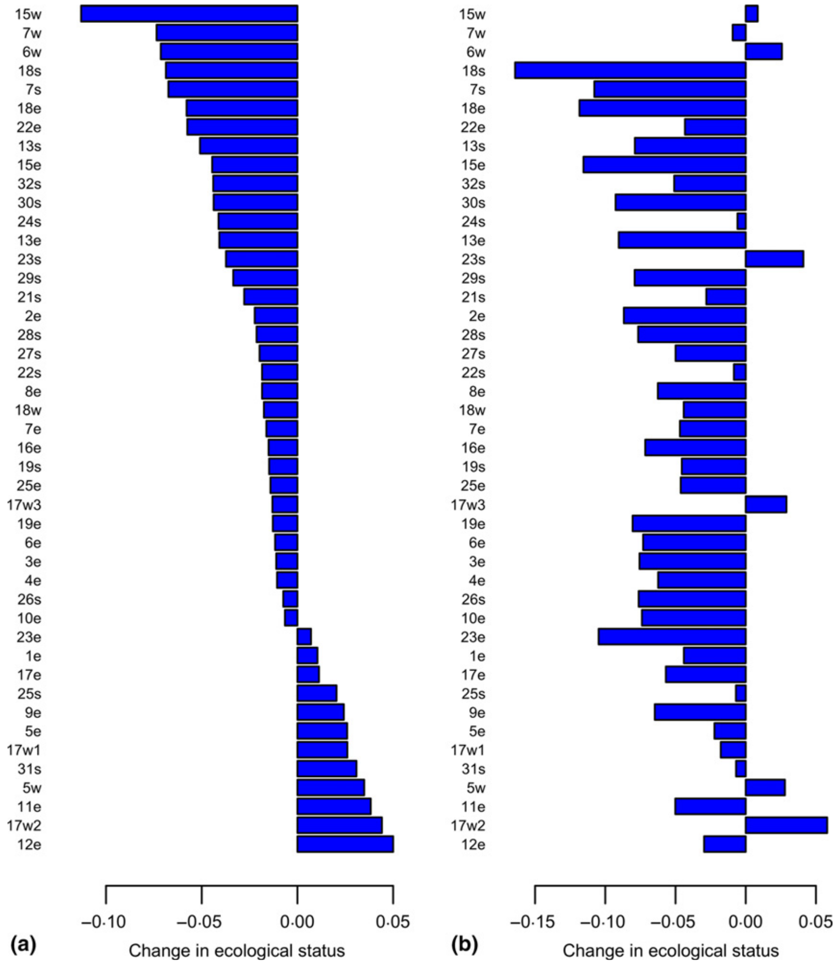

Fig. 4. Temporal change in ecological status of biodiversity in GB between the time periods, 1970–1990 and 2000–2013. Panel (a) barchart showing changes in mean ecological status of wider biodiversity (5553 species) in GB for each environmental zone over time. (b) Barchart showing mean ecological status of priority species for each environmental zone over time. For a description and location of environmental zones, see Table S1 and Figure S1.

CASE STUDY: ECOLOGICAL STATUS OF THE AREA OFFERED FOR SHALE GAS LICENCES

The mean ecological status of the area recently licensed for shale gas exploration was 0·74, and 65·3% of the hectads in this area had an ecological status that was equal to or above the mean GB value of 0·70 ± 0·002 (Fig. 5). A higher proportion of this area had 'high' ecological status (26·9%) compared to GB wide analysis (19·3%; proportion test: $\chi^2 = 28{\cdot}8$, $P < 0{\cdot}001$), and a lower proportion of the area had 'low' ecological status (6·1%) compared to GB wide analysis (18·3%; $\chi^2 = 100{\cdot}0$, $P < 0{\cdot}001$). The mean ecological status of priority species within this area was 0·74 ± 0·002 (compared to 0·74 across GB), and 26·5% of hectads had 'high' ecological status (compared to 30·9% across GB; $\chi^2 = 7{\cdot}9$, $P = 0{\cdot}004$), whilst 5% of the hectads had 'low' ecological status (compared to 8·9% across GB; $\chi^2 = 18{\cdot}0$, $P < 0{\cdot}001$).

It is also noteworthy that the area offered for shale gas licences consisted of 43 out of the 45 GB environmental zones (ITE land classes), but not all were represented equally; for example, the area constituted over 70% of the total GB coverage for nine of the zones (10e, 13e, 6e, 2e, 1e, 5e, 9e, 15e and 18e), whilst it constituted less than 10% for zones 17w1, 21s, 23s, 24s, 29s and 30s. Notably, 85% of environmental zone 5e is included in the area opened for licensing, and the proportion of zone 5e hectads with 'high' ecological status in this area was 76%.

Discussion

We have presented a biodiversity-based indicator for the quantification and prioritization of biodiversity over large spatial scales (e.g. to inform policy and development on a national scale), applying it as an example to the land offered for shale gas extraction licences in GB (see Department of Energy and Climate Change (DECC) 2016). We present it as a tool that extends the capacity of the biodiversity impact assessments currently implemented in UK Environmental Impact Assessments and Strategic Environmental Assessments. Current spatial indicators of biodiversity primarily comprise threatened species and habitats, which are often effective for impact assessment at localized scales (e.g. Treweek 2004) but are not feasible at higher spatial scales (see Rajvanshi, Mathur & Slootweg 2009). The method described here provides a more reflective indicator of large-scale biodiversity patterns, using readily available species occurrence data from hectads across an entire country. The relative measure of species richness is calculated by comparison of species

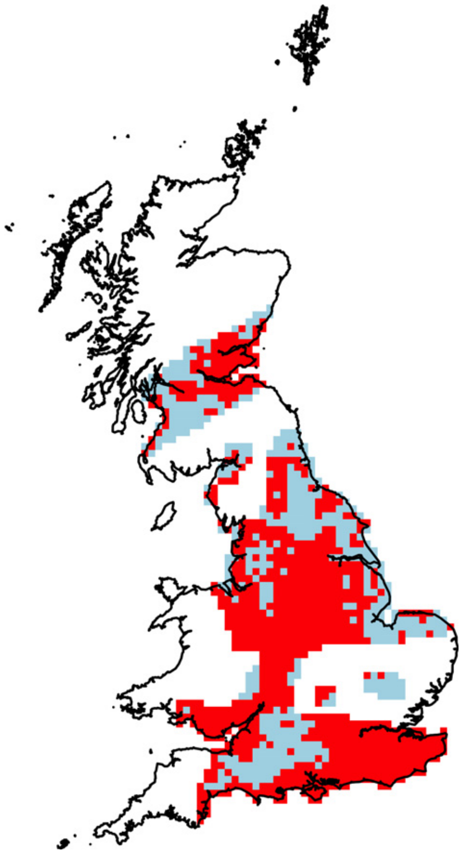

Fig. 5. Assessment of biodiversity for the area proposed for shale gas licensing. The area proposed for shale gas licences is coloured on the map: hectads with ecological status above and below the mean GB ecological status (0·7 ± 0·002) are coloured in red and blue, respectively.

richness amongst areas under similar abiotic conditions, effectively putting local biodiversity in the context of wider biodiversity.

Comparison of the ecological status derived from priority species (Joint Nature Conservation Committee 2012) with that from wider biodiversity showed that whilst there is generally a strong positive correlation, in some cases, the species richness of priority species can misrepresent the ecological status of wider biodiversity. In addition, temporal patterns between the two indicators show distinct trends. The ecological status of wider species diversity in the GB between 1970 and 2013 has increased in some areas and decreased in others, whist priority species showed a marked decrease in ecological status in all areas over this time period. This result is somewhat intuitive as priority species are identified on the basis of severe historical declines. It is notable, however, that these trends are not always reflective of trends in non-priority species, which are increasingly under threat themselves (see UK State of Nature Report 2013; Burns *et al.* 2013) and provide essential ecosystem functions and services (Balvanera *et al.* 2006; Cardinale *et al.* 2012; Winfree *et al.* 2015). Moreover, the use of priority species to assess impact over larger spatial scales (Bakken *et al.* 2014) may be misleading as they may not convey the true status of biodiversity and result in misleading estimations of impact. For example, hectad NS23 in GB has a relatively low ecological status based on priority species but higher than average ecological status based on wider biodiversity (Fig. S3). This highlights the limits of priority species and the potential advantages of using the wider species occurrence records as an indicator of the ecological status of biodiversity over wider spatial scales.

Our assessment of GB biodiversity has indicated significant spatial and temporal variation in the ecological status of priority species and wider biodiversity. These patterns highlight the potential applications of the tool presented here, which include impact assessment of development, the prioritization of areas for conservation and the identification of areas for restoration. These latter points are key components of the 2020 ‘Aichi Targets’ under the international Convention for Biological Diversity and, in the context of GB, will be essential to fulfil policy aims laid out in the government’s Natural Environment White Paper (HM Government 2011). The stratification of data according to abiotic conditions (environmental zones based on the 45 ITE land classes) means that impact can be assessed over large spatial scales (across zones) and small spatial scales (within zones), whilst accounting for large-scale relationships between abiotic conditions and biodiversity. Areas for prioritization of conservation effort could include areas of high ecological status (see Fig. 2b), or whole environmental zones with a higher proportion of ‘high’ ecological status (e.g. Zone 11e; Fig. 3a). Environmental zones with a disproportionate area of ‘low’ status as areas could be targeted as areas where development would have lower impact or, alternatively, they may targeted for potential restoration (e.g. Zone 2e; Fig. 3b), including identification of areas where connectivity may be improved to create larger areas of high ecological status. In comparing across whole environmental zones however, it should be noted that zones may also differ in the ecological uniqueness of species assemblages and it may also be desirable to include this quality in prioritization. In addition, adequate biological records data were not available prior to our 1970 baseline, and if degradation of biodiversity prior to the baseline has been uneven across zones, then they may be starting from different absolute levels of ecological status. Temporal analysis of wider biodiversity showed varying trends in ecological status across environmental zones (Fig. 4) and may help target areas for restoration and/or give indications of areas that have responded well to pressures on land use. It should be noted, however, that a wide range of socio-economic, hydrological and geological factors need to be considered in the decision of where to prioritize land for restoration versus potential development. The method presented here only provides information on current and past ecological status, which is one part of the evidence necessary in making such land-use decisions. Application of our method to the area proposed for new UK shale gas licences highlights the value in higher-level environmental impact assessments before committing to detailed infrastructure and development planning and associated local environmental impact assessments. Our results show that a large proportion of the area open for shale gas licences is of 'high' ecological status and may be regarded as important for biodiversity conservation. The area also includes large proportions of individual environmental zones. This is important with respect to potentially balancing biodiversity conservation and restoration across environmental zones to protect ecologically distinct biodiversity and provide locally available biodiversity-associated cultural services for public benefit. For example, the majority of zone 5e (85%) currently falls within the area open for shale gas licences. The combination of our approach with traditional biodiversity impact assessment of this area (i.e. threatened species, threatened habitats and land under statutory protection; see Moore, Beresford & Gove 2014) may give a far more stringent foundation on which to make predictions of impact to biodiversity. In a worst-case scenario, the failure to apply a preliminary large-scale assessment might be that the discovery of large deposits would provision large financial incentive to extract gas and oil regardless of the true value of those areas for biodiversity. Hence, there may be consequences in bypassing environmental assessments at large spatial scales, and we hope that this tool may be useful in the preliminary prioritization stages of planning processes for large-scale infrastructure and development projects.

A limitation of the method is that we do not know what the species richness was before 1970, and therefore, the maximum species richness values might underestimate the potential species richness of each environmental zone. However, we can only establish a benchmark for the earliest period at which we have suitable data, and thus, we have now established a 1970–1990 GB benchmark for considering changes in ecological status in the future. Further work could involve the development of the method to include more taxonomic groups and to reproduce the indicator at a finer spatial resolution. The 11 taxonomic groups that we use are well recorded (i.e. expected to provide robust estimations of species richness) at the hectad level and provide a good representative taxonomic sample that goes far beyond sampling in previous prioritization indicators (Moilanen *et al.* 2005; Franco *et al.* 2009). Application of this method at the 2 km x 2 km grid square scale would increase the spatial precision of species richness estimates and habitat designation (ITE class), allowing for higher spatial resolution prioritization of land use. However, it is likely to be possible to run the analyses at a finer resolution for some of the better sampled groups (e.g. birds and butterflies), but these alone are unlikely to be representative of wider biodiversity (Eglington, Noble & Fuller 2012). Therefore, a representative finer scale indicator would currently require a higher level of recording for other taxonomic groups.

In summary, we present an empirically derived biodiversity-based indicator for use in the preliminary stage assessment of ecological status at large spatial scales. This new indicator advances previous assessment tools (which rely on a limited set of indicator species), through the incorporation of large data sets of species occurrence records to give an indication of the ecological status of wider biodiversity. The method therefore provides an additional tool for the impact assessment of development, the prioritization of areas for conservation and the identification of areas for restoration at large spatial scales. The application of this method to a national-scale project highlights its potential importance for use as a preliminary stage assessment tool. Our biodiversity-based indicator has also established a baseline for the quantification of temporal trends in biodiversity across each hectad in GB, allowing future changes to be assessed relative to this. We hope that these applications will assist in the conservation and preservation of biodiversity for its own sake, as well as its role in underpinning the well being of current and future generations.

Acknowledgements

We are indebted to the national schemes and societies who have contributed species records to these analyses. They include the Bees, Wasps and Ants Recording Society, Botanical Society of the British Isles, British Bryological Society, British Myriapod and Isopod Group (Non-marine Isopoda Recording Scheme), British Trust for Ornithology, Butterfly Conservation, Dipterists Forum (Hoverfly Recording Scheme), Ground Beetle Recording Scheme, Ladybird Recording Scheme, National Moth Recording Scheme, and the Orthoptera Recording Scheme. This project was part-funded by the UK Natural Environment Research Council under the Tellus South West Project (http://www.tellusgb.ac.uk/) and a partnership of the Joint Nature Conservation Committee and the Centre for Ecology & Hydrology supporting the Biological Records Centre.

Data accessibility

ITE land classification 2007: http://doi.org/10.5285/5f0605e4-aa2a-48ab-b47c-bf5510823e8f (Bunce *et al.* 2007).

Ecological status dataset: http://doi.org/10.5285/58b248a8-6e34-4ffb-ae32-3744566399a2 (Dyer & Oliver 2014).

Estimated species richness data: http://doi.org/10.5285/6c535793-034d-4c4f-8a00-497315e7d689 (Dyer & Oliver 2016).

Species occurrence records: https://data.nbn.org.uk/.

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Received 22 April 2016; accepted 7 September 2016
Handling Editor: Ben Collen

Supporting Information

Additional Supporting Information may be found in the online version of this article.

Fig. S1. Map showing the dominant ITE land class per hectad taken from 2007 ITE land classification (Bunce *et al.* 1996).

Fig. S2. Estimates of species richness from Frescalo validated using independent monitoring data.

Fig. S3. Plot showing the relationship between the ecological status of priority species and the mean ecological status calculated from the wider biodiversity.

Table S1. Summary of land classes taken from 2007 ITE land classification (Bunce *et al.* 1996).

Table S2. Summary of ecological status data for a hectad showing a large increase in ecological status over time.

Table S3. Summary of ecological status data for a hectad showing a large decrease in ecological status over time.